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LITERATURE REVIEW AND POSITION PAPER ON COMPOSTING AS A METHOD OF REMEDIATING PETROLEUM CONTAMINATED SOILS

FINAL REPORT

SUBMITTED TO ALBERTA ENVIRONMENTAL PROTECTION

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I. INTRODUCTION

1. Summary Introduction of Composting as a Remediation Process

- **General description of composting**

Composting as a waste treatment process has evolved tremendously over the past 30 years, in reference to both our understanding of the process and the range of applications (Miller 1996a). Composting is a biological process carried out mainly under aerobic conditions in a matric phase environment. Matric phase is a soil science term and refers to an environment comprised of solids and water and air filled void spaces between solid particles. Composting systems are characterized by substrate densities sufficiently high and thermal conductivity's sufficiently low that the heat of metabolic activity causes the composting matrix to increase in temperature. Composting as a process is distinct from compost, a product. Composting has many applications beyond making compost.

In practice, process management in composting is based on the control of chemical and physical factors such as nutrients, moisture, oxygen concentration, and heat management to control temperature. Composting systems are dynamic. In a composting pile, microbial activity can use up all the oxygen, raise the temperature to lethal limits, and evaporate away the moisture, which represses positive activity.

Composting has some unique characteristics which make it particularly well suited to bioremediation. Composting ecosystems are complex and require knowledge and expertise in process management. Gradients of oxygen, pH, redox potential, dissolved substrates and other factors can exist both within particles and on the macro level, all of which effects the ecological system. Temperature and moisture gradients are common at the macro level. Ecologically, the existence of three phases and various gradients within the composting environment means that composting environments offer a very broad and complex range of available niches.

The factors of high metabolic diversity, high metabolic rates, high oxygen availability, and increased substrate availability make composting ecosystems highly aggressive systems for biological decomposition of organic materials.

Composting can also be viewed as an accelerated humification process, overcoming environmental stresses to the soil and increasing fertility by increasing soil organic matter.

- **Composting as a method of treatment for hydrocarbons**

Composting, especially with active ventilation, appears to enhance total degradation of a wide range of more difficult to degrade compounds. This is important considering petroleum contains billions of isomerically different compounds including many with complex structure and high molecular weight (Cuddington and Lowther 1977). Finstein et al. (1987) compared composting. versus simulated land farm treatment for an oil refinery sludge and found for most PAH's, 35 days of moderately thermophilic composting gave non recovery values that compared favorably with 20 days and even 1280 days of land farming treatment. In laboratory trials of land farming and composting of petroleum exploration and production waste, McMillen and Gray (1994) reported much more rapid non-recovery extractable hydrocarbons in the composting process, indicating increased hydrocarbon utilization during composting. In treatment of petroleum contaminated drill cuttings in shallow (20 - 28 cm) treatment cells, Aasen et al. (1996) reported greater hydrocarbon loss and significantly less new organic matter formation in aerated versus non aerated cells.

Liu and Sulflita (1993) remarked that bioremediation exploits the genetic diversity and metabolic versatility of microorganisms. Enhancement of native populations can lead to a broad ability to degrade petroleum in soils (Vecchioli et al. 1990). A succession of microorganisms can be viewed as an energetically efficient response to recover the greatest amount of available energy from a substrate resource (Slater and Lovatt 1984). Compared to other ecosystems, composting systems tend to offer an extreme array of ecological niches promoting metabolic diversity, coupled with high rates of overall metabolic activity (Miller 1993a). Complex substrates are often not fully degradable by a single organism, but are degraded by consortium of organisms, each carrying out different degradative steps (Floodgate 1984). Mixed populations are required to significantly degrade crude oil (Morris et al. 1996). Observations on co-metabolism and consortative decomposition would indicate that fostering both diverse and large populations would be beneficial in the decomposition of complex petroleum products.

Cerniglia (1984) compiled a listing of microorganisms reported to metabolize aromatic hydrocarbons. Bossert and Bartha (1984) compiled a listing of various genera of hydrocarbon degrading microorganism which have been isolated from soils. Generally, a broad range of bacteria, actinomycetes, and fungi can carry out petroleum hydrocarbon degradation. Miller (1996a) compiled tables of microorganisms identified from composting materials which also includes a broad range of bacteria, actinomycetes, and fungi. Of note, there is a good deal of overlap between populations of hydrocarbon degraders and organisms recovered from composting environments.

Currently, the Composting Technology Centre is working with many leading companies such as Chevron Canada Resources, Imperial Oil, and Exxon Production Research in applied research and technology transfer in this area.

2. Issues in Composting Based Bioremediation

- **Soil hydrophobicity**

Soil hydrophobicity can be a severe problem in the remediation of petroleum-contaminated soils even after petroleum hydrocarbons have been significantly reduced and toxicity is no longer observable (Yeung et al. 1997). Yeung et al. (1997) found that bioremediation of a petroleum contaminated soil had little effect on hydrophobicity. Hydrophobicity significantly decreases soil water holding capacity. Hydrophobicity also causes soils to be very difficult to rewet once drying has occurred (King 1981). After remediation, hydrophobic soils make the establishment of plant growth difficult, because the first time the field soil dries out vegetation dies from lack of water.

Addition of compost to remediated soils might be a worthwhile means of overcoming hydrophobicity and establishing plant cover. Compost, by increasing soil organic matter, can increase soil aggregation and greatly increase water holding capacity, thereby mitigating hydrophobicity. Establishment of plant cover could further decomposition. While phytoremediation appears promising there is a lack of basic research in this area (ES&T 1997).

While petroleum hydrocarbon induces oil hydrophobicity and is recognized as a problem, it is not clear how much actual research to mitigate this problem has been carried out.

- **Bioavailability**

Bioavailability is an important technical and regulatory issue.

Bioavailability is the major reason that biological decomposition of petroleum in soils can take years to occur. Bioavailability is determined by various chemical, physical, physiological, and ecological factors (McGill et al. 1981). Generally, the n-alkanes and aromatics are the most biodegradable, followed by the branched alkanes and cycloalkanes (Bossert and Bartha 1984). Of the various petroleum fractions, polycyclic aromatic hydrocarbons (PAH's) appear to be the most studied because of potential hazard (Cerniglia 1981, Cerniglia 1993, Menzie et al. 1992), and environmental persistence (Gibson and Subramanian, 1984). Therefore, biodegradation studies of PAHs tend to predominate the literature over other petroleum components.

Bioavailability can be limited by sequestration when petroleum fractions are added to a soil or compost (Means et al. 1980). Clays can oligomerise PAHs into unextractable forms with low bioavailability; this can happen as quickly as a 100% loss of recovery within a few weeks (Karimi-Lotfabad et al. 1996).
PAHs can also form complexes with organic matter, which is unextractable and poorly available (Hogan 1995). Work by Douglas et al. (1996) further demonstrated that weathering, including soil complexing, significantly decreases petroleum hydrocarbon bioavailability.

Bioavailability can also be directly affected by microbial activity. Much has been made of the abilities of white rot fungi to degrade a range of compounds that are resistant to microbial degradation (Reddy 1995). White rot fungi can degrade complex compounds through non-selective enzyme mediated free radical reactions (Barr and Aust 1994). Yet, these same free radical reactions can also cause polymerization reaction which bind compounds into soil organic matter through humification (Qui and McFarland 1991). Molecules, which have been humified, have very low bioavailability and slow turn over rates, and are not likely to be re-released in their original state (Stevenson 1982). Humification is an interesting regulatory issue, in that mineralization does not occur but yet environmental hazard is greatly reduced.

Diepersants can greatly increase oil decomposition by improving bioavailability (Atlas and Bartha 1973, 1992), and are of benefit in soil bioremediation (Grimberg et al. 1996, Sabatini et al. 1996, Shiau et al. 1996, Salma and Miller 1996), although not all surfactant treatments are of benefit (Wilson et al. 1996). Alternately, a great number of surfactant treatments are of benefit (Wilson et al. 1996). Alternately, a great number of microorganisms can produce biosurfactants, which can greatly increase bioavailability of petroleum components (Zajic and Mahemedy 1984, Deziel et al. 1996), and such organisms would grow during composting. An argument against adding surfactants during composting is that common detergents are rapidly broken down during composting (Robert Dow, personal communication).

In composting systems amended with organic nutrients ammonification might affect bioavailability. Ammonia can solubilize organic matter, and also chemically react with a broad range of compounds increasing bioavailability (Miller 1993). Whether or not ammonia would effect petroleum bioavailability appears to be unknown.

Bioavailability can also be limited by physical constraints such as gas exchange, water availability, and physical barriers to microbial colonization. The effect of crude oil on soil structure was investigated by McNabb et al. (1993). It was found that contaminating the soil pore fluid with crude oil led to: higher soil water repellency, less resistance to deformation, higher plasticity, lower water holding capacity, and less stable aggregates. These changes adversely decrease pore space and therefore gas exchange.

- **Moisture**

Generally, optimum microbial activity is achieved by the maximum water content that does not restrict oxygen diffusion. Water is crucial to transport of many microbial cells and therefore substrate colonization (Griffin 1981; Harris 1981, Miller 1989). In land farming of petroleum wastes Dibble and Bartha (1979) found 30 - 90% of soil moisture capacity was optimal. Composting petroleum contaminated soil (1.7% oil and grease) with added leaves and alfalfa in a laboratory system, Beaudin et al. (1996) maintained moisture within a range of 50 - 60%. McMillen and Gray (1994) composted a oily sludge with added tree waste, horse manure and soil, with a total mix extractable hydrocarbon content of 5.94% and found 39% moisture good. With very lightly amended Diesel invert mud and cuttings, biodegradation studies by Danielson (1994) worked with a moisture content of 20 - 22%. A matric potential of -10kPa is optimal for composting activity (Miller 1989). A potential of -10kPa might represent 60-70% moisture for organic materials but perhaps around 30% for field soils.

Wiley (1957) noted that any oils that are liquid at composting temperatures should be added to the gravimetric water content, because liquid oils and water have the same effect on gas exchange. Wiley (1957) considered oil and water together as the ‘percent liquid’.

- **Temperature**

Temperature control is a critical management issue in composting. Biological systems in general exhibit the 'Q10 effect', in that for every 10 °C increase in temperature activity doubles. Of course, microorganisms have lethal temperature limits at which activity stops.

Much available information related to optimal temperatures for biodegradation of petroleum fractions comes from land farming situations. In the composting of plant or animal derived materials, the rate of composting increases with temperature until about 60 °C. Above 60 °C, activity rates drop rapidly with additional increases in temperature (MacGregor et al., 1981, Finstein et al., 1983, Finstein et al., 1985; Kuter et al., 1985; McKinely, and Vestal, 1984; McKinely and Vestal, 1985). Activity rates at 70 °C are about 1/10 the rates at 60 °C, at about 80-82 °C microbial activity completely stops (Miller 1993).

In land farming the highest degradation rates are in the 30-40 °C range (Bossert et al. 1994, Bossert and Bartha 1984). Bioremediating Diesel invert mud and cuttings Danielson (1994) found decomposition rates were highest at 30 and 35 °C but only if an adapted seed was used. Atlas and Bartha (1992) felt that at temperatures over 30 °C biodegradation of petroleum would decrease. Composting mineral oil contaminated soil, Beaudin et al. (1996) kept temperatures at 50 °C. McMillen and Gray (1994) composted oily wastes at 45 °C. Hogan et al. (1989) composted representative aliphatic and polyaromatic hydrocarbons at both 50 °C and 35 °C, and found that after 35 days the residual concentrations of the various compounds at 35 °C were 1/2 the values in of the 50 °C treatment.

Temperature as a factor in PAH degradation was considered definitively in the investigations of Hogan (1995) composting creosote. Hogan composting a range of PAH's at temperatures of 35, 40, 45, 50, 55, and 60 °C. In general, the smaller PAH's appeared to compost well over the range of 35 - 55 degrees Celsius. For the heavier PAH's, over a 20 day decomposition period the 45 °C temperature consistently gave the greatest total loss (as non recovery) of the initial PAH concentration. An exception was chrysene, which was lost most rapidly at 60 °C. This behavior of chrysene was also observed by Finstein et al. (1987).

Some oil degrading Bacillus with temperature optimum of 60 °C have been reported (Sorkhoh et al. 1993) but most oil degrading bacteria, such as the strains listed Bossert and Bartha (1984) would have lower optimal temperatures. A fair number of recent papers have considered white rot fungi which decompose of PAH's (Bogan et al, 1996a, Bogan et al, 1996b, Lestan and Lamar 1996, Bogan and Lamar 1996, Collett 1992, Bezalel et al. 1996abc). Lonergan et al. (1994) found that for a number of different white rot fungi, most had restricted growth below 20 - 25 °C, with maximum growth rates occurring at anywhere between 35 - 45 °C.

With the usage of organic amendments in appreciable amounts to carry out soil composting, formal temperature control would be necessary to avoid exceeding the upper lethal limit for the petroleum degrading populations. The tendency for composting systems to overheat until biological activity comes to a halt is intrinsic to such systems (Miller 1993a).

- **Oxygen**

Oxygen is important to the degradation of petroleum factions as they tend to be highly reduced. Bossert and Bartha 1984) reviewed a number of studies of oil decomposition in soils, and oil degradation rates were always highest when aeration was maximized. In research with Diesel invert mud and cuttings, Aasen et al. (1996) reckon from their results that oxygen concentrations lower than 10 - 16% could slow the rate of hydrocarbon biodegradation. Under otherwise optimal composting conditions, decomposition rates have been observed to slow at less than 12 - 14% (Miller et al. 1990, Harper et al. 1992). Regardless of interstitial oxygen concentrations, if aggregates being treated are overly large or wet, serious oxygen limitations will occur.

In the research of Aasen et al. (1996), the use of forced aeration significantly increased the mineralization of petroleum contaminated drill cutting compared to non-aerated but interstitially well oxygenated soil treatments. The ventilation volocity independent of interstitial oxygen concentration can significantly increase decomposition has been observed in waste composting and might be related to turbulence aided diffusion at the particle level (Miller 1996c).

- **Inorganic nutrients**

While petroleum can supply a good deal of metabolic carbon, petroleum is deficient in other nutrients, such as nitrogen and phosphorus. Suitable ratios of C:N:P are not readily apparent for petroleum degradation, however, in that much of the carbon in petroleum is of limited availability. McMillen and Gray (1994) found a standard C:N:P ratio of 100:5:1.7 worked well for degrading petroleum wastes. Generally, reports of specific C:N:P ratios are lacking in the literature, and effects of N and P addition are at times conflicting (Bossert and Bartha 1984). For Diesel invert muds and cuttings, Danielson (1994) found that a C/N ratio of 100:1 worked well and that higher ratios only led to undesirable N losses. Under long decomposition period's bacteria, and especially fungi, recycle nitrogen very effectively (Alexander 1977). Best ratios for C:N:P are likely to be contaminant specific and related to carbon availability. A slowly available carbon source can be degraded with wider C:N:P ratios because N and P will be recycled during decomposition.

- **Organic nutrients agents**

Addition of significant amounts of plant materials and manures to petroleum contaminated sub-soils can be used to initiate a 'soil composting' process.
Advantages of soil composting include;
a.)significantly increasing the populations of saprophytic microorganisms by providing a readily available carbon source and provision of the required macro and micro-nutrients in organic form.
b.)improvement of physical structure for decomposition by reducing bulk density and increasing porosity.
c.) provide sufficient nutrition to increase temperatures into favorable ranges for sustained periods.

One issue observed by Hogan (1995) was that for some otherwise readily degradable PAHs, nutrient addition can, on the short term, spare these compounds from degradation.

This observation is not unlike that made by Somasundaram et al. (1987) who observed that for a range of soil applied insecticides, soil manure application resulted increased persistence of some otherwise biodegradable pesticides. At the same time, various substrates can induce expression of enzymes that in turn can metabolize other substrates (Clarke 1984). Various substrates might also select for genetic changes that affect metabolic versatility (Liu and Suflita 1994). Adding to this complexity, the physical environment of a composting matrix can also be a powerful inducer of degradative enzymes (Miller 1996b). With the complexities of composting ecology and petroleum composition, induction and repression of degradation related to nutrition and other factors requires further research.

- **Salinity**

Addressing the salts issue often associated with hydrocarbon contaminated sites and drill cuttings is sometimes overlooked. When assessing any remediation approach, the fate of the salts should be addressed.

Salinity of drill cuttings can be high based on the nature of underlying ground water. A value greater than 4 ms/cm would denote a saline soil where salt content would seriously interfere with the growth of most plants (Brady 1974). Danielson (1994) reports that the soluble salt content of Diesel invert muds and cuttings from the Albert Slopes region can be high (37 ms/cm) but can be readily removed by water leaching before bioremediation of the oil fraction.

- **Soil texture and class effects in remediation**

Soil type, including texture and classification, is an important consideration in bioremediation. Soil texture, i.e. the particle size distribution of the soil, has a great effect on structure and activity. Amount of clay in a soil is very important in bioremediation because of the effect of clays on soil porosity and more complex chemical aspects. Clays can catalyze humic acid formation and protect organic material from decomposition within aggregates (Stott and Martin 1990). The reported half lives of 1,700 to 2,200 years for humus in some Canadian soils (Paul et al. 1964) is related not only to the resistance of humus to decomposition, but also to soil texture and class effects and protection by clays.

Soil type is of interest from a regulatory standpoint, as soil type has a direct bearing on remediation progress and achievable end points.

- **Use of special inoculant cultures**

In reference to the role of specific populations in petroleum remediation, seeding with specific cultures for bioremediation of petroleum products has not been of value to date in natural environments (Atlas and Bartha 1992). Inoculation of petroleum contaminated invert mud residues with commercial cultures for petroleum remediation demonstrated no benefit in trials by Danielson (1994). On the other hand, use of adapted seed cultures is a very useful method with a long history of application in biological waste treatment. Danielson (1994) did find that the use of an adapted seed culture significantly increased petroleum decomposition, and especially when treatment temperatures (35 °C) were higher than the normal soil ambient temperatures.

3. How the search for information was carried out

The literature review contained herein was intended to be a broad and comprehensive consideration of recent literature related to soil composting as a method for remediating petroleum contamination of soils. Key issues guiding this review are further understanding; a) the physical and biological processes involved in composting, and b) how the properties of composted materials affect its end use.

The literature was searched in the following manner. The primary means of search was that of a title search of all potentially relevant journals in the Agriculture and Environmental Sciences edition of Current Contents for the period of January 1, 1997, through June 30, 1998. Copies of relevant titled papers were obtained from university library collections for almost all titles identified, others are from more obscure journals. From the title search, the most active relevant researchers were identified, and letters to them describing the review were sent out requesting paper reprints and any other relevant information. Citations from recent papers were also reviewed to identify other recent papers of note. Computer website searches were also carried out looking for additional documentation and information. A large number of governmental and non governmental websites were searched, including major sites such as the USEPA, the USDA National Agricultural Library, various military environmental sites, sites of the Canadian Association of Petroleum Producers and the American Petroleum Institute, the Petroleum Environmental Research Forum, and various corporate sites. Generally, title searching and correspondence with the most relevant active researchers provided the most extensive information, while the computer websites and databases provided the least useful information.

While there is a good deal of literature with relevance to the soil composting issue, there is a rather limited literature directly based on applied soil composting. There has been a good deal of soil composting work carried out by or funded by the petroleum industry, but most of this work has been carried out for internal information purposes and is not in the open literature. A number of researchers contacted during the review related that they had carried out industry funded projects, but that the results of such projects were proprietary and never published.

II. REVIEW OF THE LITERATURE

II A. Physical and Biological Processes Involved in Composting

1. Biological treatment of petroleum contaminated soils including composting and related landfarming methods

The extent of current projects using a soil composting method for remediation of petroleum contamination is difficult to ascertain, although anecdotal information from academic scientists and petroleum industry researchers indicate that a significant amount of pilot and full scale work is being carried out. Yet, for a number of reasons, there is a paucity of publications reporting the results of large scale composting projects.

In the journal literature only one paper, by Al-Daher et al. (1998), reports specifically on the use of turned composting soil piles for the remediation of petroleum contaminated soils. Al-Daher et al. conducted a pilot scale composting project in late 1992 in the Burgan oil field site in Kuwait to treat soils contaminated during the Gulf War. Sixteen different treatment designs were implemented in 20 m3 size windrows. The sandy contaminated soil was amended with fertilizer (NPK 15:15:15) to give a C:N ratio of 75:1 or 50:1, and all piles were amended with woodchips as a bulking agent. Piles were turned and wetted once a month with a front-end loader for the eight-month trial period. For the trials various combinations of amendment with mature compost, dried sewage sludge, and adapted microbial inoculant were implemented. After three months the effects of microbial inoculant were insignificant, and there was no significant differences between any of the treatments at the end of the trial period. Overall reduction in petroleum hydrocarbons was 54% for lightly (11 mg/kg) contaminated soil and 60% for heavily (15 mg/kg) contaminated soil. A major conclusion of the study was that maintaining moisture in the piles was very important. Lacking in this paper is any ecological information on the environmental processing conditions in the windrows.

Soil composting was successfully used to remediate petroleum hydrocarbon contaminated soil at the Dubose Oil Products Co. US EPA Superfund site in Florida (US EPA 1998). Soil treatment was carried out in contained bed of a little over one metre in depth, with a vacuum ventilation system underneath the beds that discharged to an activated carbon air scrubber. Ventilation was controlled to maintain a 20% interstitial oxygen content. Treatment was carried out over a 6 month period. Soil was amended with N:P:K to 120:10:2 and a moisture content of 15%, and inoculated over a two day period with an adapted seed inoculant derived from native soil. This treatment reduced initial PAH's of 50.8 to 576.2 mg/kg down to 3.3 to 49.9 mg/kg. Treatment costs were given as US $266 per ton treated, or US $90 per ton excavated. No ecological information on processing conditions are provided in this report.

Bleckmann et al. (1989, 1997) reported on the long term results of landfarming of crude oil contaminated production sands in Oklahoma. Over a 13 year period biodegradation processes removed over 80% of applied oil, and that the half life of the oil in the treated soil was approximately 3 years with significant variation between years. This was notably longer than the 50 to 150 day half lives range for refinery wastes and sludges published by the API (1987). Over time there was a slight increase in the proportions of resins and asphaltenes. Compared to the Al-Daher et al. (1998) and EPA (1997) findings, oil degradation during landfarming was slower than composting.

In pilot scale studies of bioremediation of petroleum contaminated soil, Cho et al. (1997b) investigated the biodegradation of various components of crude oil and the microbial populations responsible for remediation. Cho et al. concluded that the saturated fraction was the most readily degradable, the aromatic fraction was degraded but only very slowly, and that the sulfur containing compounds were relatively resistant. An interesting conclusion was that aromatic compounds were only slowly degraded in Kuwaiti soil, because the very low soil moistures common to Kuwaiti soils disfavor the gram negative bacteria most adept at degrading aromatics.

Margesin and Schinner (1997a) in laboratory experiments investigated the bioremediation of diesel oil contaminated alpine soils at low temperatures. They found that cold adapted indigenous microbial populations were quite able to degrade diesel at 10°C, and that the use of special inoculum adapted to cold and diesel enhanced degradation only slightly and temporarily. 4,000 mg/kg of diesel was reduced to 380-400 mg/kg in 155 days, with 30% of the diesel lost abiotically and 60% to microbial degradation. In a second study (Margesin and Schinner 1997b) found that after 30 days 43% of diesel oil was degraded at 10°C, while 55% was degraded at 25°C.

Two technologies that could potentially compete with soil composting methods were reported. In pilot scale, Pradhan et al. (1997) investigated a process called MGP-REM to treat PAH contaminated soils. The MGP-REM process uses a slurry phase bioreactor, followed by a chemical reactor using Fenton' reagent (H_2O_2 and ferrous ions ($Fe2+$)), and then further biological treatment. PAH degradation was much higher in the MGP-REM process than in conventional biological (slurry) treatment with high removal rates of even 5 and 6 ring PAHs. Costs for the MGP-REM process were given as US $100-135 per cubic yard of soil. Masten and Davis (1997) reviewed a process of in-situ ozonation. In-situ ozonation has been reported to rapidly (hours) reduce PAHs significantly. While considering various application factors, Masten and Davis reckoned that in-situ ozonation is likely to be cost competitive with other soil remediation technologies (typically US $100 per ton).

Worthy of comment is what is not in the literature. For all the interest in soil bioremediation, the lack of published field or pilot scale studies is conspicuous. For the soil composting that has been published, there is an almost total lack of ecological information related to process control. In various other applications of composting, the effect of various environmental parameters are known to have a very great effect on microbial population development and activity (Miller 1993). Yet, in soil composting, process management is appears to be by default, and reported studies offer so little information that an ecological consideration of process management is not possible. In the field studies of Al-Daher et al. (1998) and the EPA (1997) not even temperature data is reported. Also apparently lacking in the literature are direct comparative studies between soil composting and other remediation methods, making comparisons between efficacy and cost rather difficult.

2. Amendments of organic matter, fertilizers and microbial inoculants

The usage of fertilizers to enhance soil biodegradation of petroleum hydrocarbons has been recognized for quite a while (Dibble and Bartha 1979), but further refinement in this area continues. Recent work reviewed here is all based on studies, which while not composting studies per se, have relevance to soil composting remediation.

Margesin and Schinner (1997c) added nitrogen at C:N ratios of 50:1 and 10:1 and found that both ratios greatly increased diesel oil degradation over non-fertilized controls. While at the 50:1 ratio the use of an adapted inocula somewhat increased degradation rate, the effect of inocula at the 10:1 ratio was almost entirely eliminated.

Braddock et al. (1997), in field trials on fuel contaminated sandy soil in Alaska, added N:P at levels of 100:45, 200:90 and 300:135 mg/kg soil and found that while the lowest addition of N:P significantly increased the loss of hydrocarbons, the higher additions did not increase loss. This was determined to be caused by the significant increase in osmotic potential with increased fertilizer application, which has been observed to inhibit the degradation of hydrocarbons. Fertilizer addition was addressed fundamentally in the work of Walworth et al. (1997a,b), in which it was determined that expressing N in units of mg N/kg soil water gave the best correlation between activity and N level. Expression of N on a soil water basis reflects both the effect of N addition on soil osmotic potential and on microbially available N in the soil solution. Based on experimentation, an optimal N level of approximately 2000 mg N/kg soil water was tentatively proposed. To avoid osmotic inhibition of bioremediation by fertilizer addition, Woolard et al. (1997) and Walworth et al. (1997c) found the use of fish bonemeal to be a good source of slow release N and P, with suitable rates of release for cold climate soils.

Kireeva et al. (1996a) carried out field and laboratory studies on the bioremediation of a heavy loamy soil contaminated with an average of 9.6% oil. Activated sludge was used as a amendment for mineral nutrients, available substrate, and microbial inoculant. Activated sludge contains microbial populations adapted to growth on oils and grease, as oils and grease typically represent more than 10% of waste water solids (Metcalf & Eddy, Inc., 1972). Compared to an unamended control, the activated sludge amended plots exhibited a much greater loss of oil, especially after two and three years. While the crop for barley was greatly diminished over all three years for the untreated oil contaminated plots, in years two and three the activated sludge amended plots demonstrated an even higher crop production capacity than uncontaminated control plots. This is of note, in that regaining soil fertility of oil contaminated soils is usually difficult.

In the biodegradation of some compounds not normally found in soils, inoculation with adapted seed cultures is a common practice. Guthrie and Pfaender (1998), in laboratory studies carried out over 9 months, found pyrene was significantly degraded in soil augmented with a pyrene degrading community, but not at all in an unamended soil. On the other hand, sites that have been contaminated with petroleum products for a long time tend to develop diverse petroleum utilizing populations (Steward et al. 1997).

3. Sequestration and bioavailability of petroleum components in soils

Many heavier petroleum fractions, including 5 and 6 ring PAHs, are biodegradable, but the rate and extent of degradation in soils can be greatly diminished because of sequestration and low bioavailability.

Hatzinger and Martin (1997) investigated the sequestration of phenanthrene into solid alkanes, waxes, some low molecular weight polymers, and in silica particles. Data was consistent with the hypotheses that partitioning into organic matter or presence in nanopores decrease bioavailability. Partitioning into silica nanopores reduced bioavailability. Of the nonporous solids, waxes and polymers greatly reduced the rate of phenanthrene mineralization, while polysaccharides did not appreciably affect mineralization. Chung and Alexander (1998) investigated the sequestration of phenanthrene and atrazine in 16 different sterilized soils. They found sequestration rates and extents varied markedly between different soils, and that there was not a good correlation between sequestration for the two compounds in the various soils. While strong extractants gave high levels of compound recovery even after 120 days of aging, weak extractants and bioassays revealed substantial sequestration. Weak extractant recoveries were not found to be good predictors of bioavailability. White et al. (1997) reported that in the sequestration of phenanthrene in different soils, cycles of wetting and drying during aging substantially increased sequestration. White et al. suggested that the use of wetting and drying cycles could reduce contaminant availability to the extent that such cycling could be used as a form of remediation. White et al. also found a lack of relationship between clay content and sequestration for the three clay soils tested.

Kanaly et al. (1997) investigated cometabolic biodegradation of radiolabelled benzo(a)pyrene. They found binding of the label on day 0 was about 4 times higher in the biotic soil than in the abiotic control, indicating the significance of biological activity in sequestration. Guthrie and Pfaender (1998) found that binding of the label from labeled pyrene into soil organic matter was about 6 times greater in biotic than in an abiotic control, i.e., microbial activity greatly contributed to binding. In two biotic soils binding of label was similar, even though mineralization of pyrene occurred only in the one soil with a adapted inoculum. These results indicate that while binding and decomposition are both mainly biotic events, binding and decomposition might be carried out by different populations.

A great deal of organic matter is added to soil during soil composting and subjected to conditions fostering humification. De Paolis and Kukkonen (1997) compared the binding capacities of humic and fulvic acids for benzo[a]pyrene and found that humic acids were more binding than fulvic acids because of the higher aromatic content and lower content of functional groups. pH had little effect on the binding of the neutral PAHs. Xing (1997) reported that the quality of soil organic matter markedly influenced the binding of naphthalene, and that old soil organic matter from shale had a higher affinity for naphthalene than younger organic matter. Xing also observed that binding of naphthalene increased with soil organic aromaticity and decreased with polar functional groups. Raber et al. (1998) noted that the partition coefficients of PAHs into natural soil dissolved organic matter is significantly different from commercial fulvic and humic acids often used in modeling studies. Raber et al. found that PAH binding to a mineral soil was considerably lower compared to acid forest floor materials, and that binding was only slightly affected by pH. Raber et al. also observed that more hydrophobic organic matter (more aromatic and less polar) had a much higher binding capacity for PAHs than more hydrophilic (more polar functional groups) organic matter. Totsche et al. (1997) examined the effect of dissolved organic matter on PAH retention in soils, and, in contrast to observations in aquifer systems, found that the organic matter greatly reduced PAH mobility.

PAHs can also bind to non-aqueous phase liquids (NAPL) in soils. In a creosote contaminated soil, Rutherford et al. (1997) found that PAHs partitioned about 4.6 to 8.3-fold more effectively into NAPL than into native soil organic matter. Comparing a creosote remediated versus non remediated soil, bioavailability of naphthalene could be predicted as a function of the remaining amount of contaminant, independent of the compositional changes which took place in the creosote during remediation. Similar observations were made by Zemanek et al. (1998) in which for petroleum and creosote contaminated soils, 84 and 95% w/w of the PAHs existed in the oil phase, with the exception of naphthalene (70%). Zemanek et al. concluded from their work that for the successful remediation of PAH contamination in soils, the focus of remediation must be the residual oil phase.

Bosma et al. (1997) quantified bioavailability by modeling mass transfer limitations in biotransformation. Their model for a threshold concentration for growth was developed to permit an estimation of; i) to what extent remediation may be hampered by either microbial activity or mass transfer limitations and (ii) whether contaminant concentrations were already below a site-specific threshold concentration. From an analysis of bioremediation data, Bosma et al. concluded that intrinsic microbial activity limits bioremediation only in a few cases, but in most cases bioremediation is limited by mass transfer limitations. Bacteria can live a famine existence even at high concentrations of degradable pollutants.

Cornelissen et al. (1998) examined the desorption kinetics of 15 PAHs from sediments before and after bioremediation in a bioreactor or landfarm. They found rapidly desorbing PAHs were degraded during bioremediation while the slowly desorbing fractions remained unchanged. The initially rapidly desorbing fraction could be used to roughly predict the extent of possible PAH degradation.

A proposition put forth by a number of researchers (Bosma et al. 1997; Chung and Alexander 1998; Hatzinger and Alexander 1997; White et al. 1997; Guthrie and Pfaender 1998) is that sequestration significantly reduces bioavailability, and therefore risk. Risk based standards for site clean up criteria based on bioavailability appears to be technically feasible. In terms of remediation costs and environmental risk, especially for sites with aged contaminants, bioavailability based standards might have more technical justification than total concentration standards based on strong extraction methods.

4. The use of surfactants in soil bioremediation

Surfactant usage in soil bioremediation is a topic of significant current research interest and has been recently reviewed (Miller 1995a). In overview, the usage of surfactants to increase the bioavailability of hydrophobic petroleum hydrocarbons is neither a simple nor well developed technology. Surfactants can increase bioavailability by greatly increasing miscibility of hydrophobic liquid and solid phase organic compounds, and also pseudosolubilize liquids into emulsions of small droplets. Yet, while the use of surfactants can increase biodegradation in some situations, surfactant use can have no effect or even decrease biodegradation in other situations. Dependent on surfactant type and concentration, surfactants can be toxic to desired microbial populations, can adversely affect microbial cell hydrophobicity (causing cells to agglomerate), interfere with cell adherence, or the surfactant can be preferred as a substrate over hydrocarbons and thereby spare them from degradation. Many surfactants will form micelles above a concentration known as the critical micelle concentration (CMC); the effects of a surfactant can be very different above and below the CMC. Generally, as the concentration of a surfactant is increased, liquid surface tension decreases until the CMC is reached beyond which no further reduction in surface tension occurs. Surfactant usage can also induce changes in microbial phenotypic expressions related to the ability to degrade specific substrates. Application of a surfactant in bioremediation requires consideration of surfactant type, structure, and concentration, substrate structure and concentration, and microbial type and numbers. Miller (1995a) states: "...the lack of knowledge concerning the mechanisms of surfactant-enhanced degradation is limiting their application to remediation technologies."

Surfactants for bioremediation can be broadly divided into synthetic surfactants and biosurfactants (Miller 1995a). Synthetic surfactants can vary greatly in structure and mechanism of action. Synthetic surfactants can be non-ionic, ionic, or amphoteric. One of the more studied groups of surfactants are the ethoxylate derivatives (Tegitol series). Biosurfactants are produced by many bacterial genera with a role in substrate degradation. Known biosurfactants are either anionic or nonionic, and can be classed as glycolipids, lipopeptides, lipopolysaccharides, phospholipids, and fatty acids/ neutral lipids. While much research has been carried out with synthetic surfactants, biosurfactants have also received considerable research attention. Research in biosurfactants has been stimulated by the perception that being naturally occurring their large scale usage in remediation will cause less environmental concern, and that better understanding of biosurfactants could provide means of stimulating microbes to produce biosurfactants in-situ.

a. Synthetic surfactants

Guha et al. (1998) investigated the effect of the presence of multiple PAHs on the surfactant induced micellar partitioning of individual compounds using Triton X-100. They found that the presence of less hydrophobic PAHs can increase the solubility of the more hydrophobic PAHs by up to 90%. Yeom et al. (1996) looked at the release of PAHs from a weathered contaminated soil using a polymeric adsorbent, Tenax-TA and a nonionic surfactant, Brij 30. Diffusional mass transfer of PAHs in the soil matrix limited PAH release, and was not affected much by Tenax-TA. Brij 30 substantially increased PAH dissolution from soil by swelling the weathered tar matrix and enhancing matrix diffusivities. In an in situ field experiment, Jawitz et al. (1998) used a Winsor Type I surfactant/ alcohol mixture (polyoxyethylene(10)oleyl ether and 1-pentanol) flood to remove a mixture of fuel, waste oil, and solvents from a contaminated soil. The flood removed 90-95% of the most common contaminants but could not remove a non-solvent extractable pitch fraction.

Churchhill and Churchhill (1997) studied the effect of the non ionic surfactants Triton X-100 and Tween-80 on the degradation of octadecane by *Pseudomonas aeruginosa, Rhodococcus erythropolis* and two strains of *Acinetobacte*r. *Acinetobacter* strains, whose cells are very hydrophobic, showed no change in their rate of mineralization of octadecane in the presence of surfactants. *P. aeruginosa* and *R. erythropolis*, whose cells are hydrophilic, increased their rate of octadecane mineralization in the presence of surfactants. Overall, the two hydrophobic *Acinetobacter* strains were much faster octadecane degraders than the other two strains. Kotterman et al. (1998) investigated the effect of Tween-80 on the ability of the white rot fungus *Bjerkandera* sp. to degrade PAHs. Tween-80 increased PAH degradation rates by two to five fold solely due to the increased bioavailability. This stimulatory effect was lost within 24 hours, however, because the fungus also rapidly mineralized the Tween-80.

Grimberg et al. (1996) investigated the degradation of phenanthrene by *Pseudomonas stutzeri* in the presence of the surfactant Tergitol NP-10. Grimberg et al. found Tergitol NP-10 significantly stimulated phenanthrene degradation, and described this in a mathematical model of the physical effects of the surfactant on partitioning of phenanthrene between aqueous and micellar phases and on dissolution of the solid hydrocarbon. Their model predicted that an overdosing of surfactant would decrease degradation because of a decline in aqueous phase hydrocarbon concentration.

That surfactant usage could both increase and decrease PAH degradation was reflected in a number of studies. Jimenez and Bartha (1996) studied the degradation of pyrene by a *Mycobacterium* strain characterized by a high cell hydrophobicity and an ability to grow solely on pyrene. Detergent below the CMC increased mineralization, but at concentrations above the CMC mineralization was severely inhibited. Polyethylene glycol and a number of hydrophobic solvents were found to inhibit degradation, while mineralization was doubled by paraffin oil, squalene, squalane, tridecylcyclohexane, and cis-9-tricosene. *Mycobacterium* cells physically adhering to solvent droplets metabolized pyrene 8.5 times faster than cells suspended in the aqueous medium. Tiehm (1994) found that the growth of mixed cultures on phenanthrene and fluoranthene was inhibited by sodium dodecyl sulfate, because the surfactant was preferred as a substrate over the PAHs. Tiehm (1994) found nonionic surfactants of the alkyleoxylate and alkylphenolethoxylate type were toxic to PAH degraders, but that toxicity decreased with increasing hydrophilicity (increasing ethoxylate chain length). In further work Tiehm et al. (1997) looked at the effect of using two alkylphenolethoxylate type surfactants, Arkopal N-300 and Sapogenat T-300, in treating PAH contaminated soil. Arkopal N-300 was biodegraded so rapidly it depleted soil oxygen and inhibited PAH degradation. Sapogenat T-300 was less degradable and increased degradation until after 54 days it too lead to oxygen depletion. Bruheim et al. (1997) investigated the oxidation of crude oil alkanes by the bacterium *Rhodococcus* sp. in the presence of Tergitol 15-S-7, Brij 30, Span 20 and Tween 85. Bruheim et al. found that all of these surfactants significantly decreased crude oil utilization by exponential growth stage *Rhodococcus* cells, but some surfactants stimulated stationary growth rate cells. Cells of *Rhodococcus* are highly hydrophobic during exponential growth and attach to oil readily, while stationary growth cells are less hydrophobic.

b. Biosurfactants

That PAH degrading bacterial strains can produce biosurfactants in the presence of PAHs has been only recent demonstrated (Deziel et al. 1996). Dezeil et al. had some evidence that the rhamnolipids (amphiphilic surface-active glycolipids) produced by *Pseudomonas aeruginosa* were induced by diffusible autoinducers generated in response to PAH degradation.

The most active research group looking at biosurfactants for bioremediation is that of Raina Miller of the University of Arizona and colleagues, producing over a dozen recent papers considered in the remainder of this section. A major point of Miller and colleagues work is that biosurfactants work by increasing hydrocarbon solubilization and changing the affinity of microbial cells and hydrocarbons by increasing cell surface hydrophobicity.

Zhang and Miller (1994) investigated the effect of rhamnolipid biosurfactant on the cell hydrophobicity of four different strains of Pseudomonas aeruginosa and the overall degradation of octadecane. Rhamnolipid increased octadecane aqueous dispersion by greater than 104, but inhibited biodegradation by all four bacterial strains for at least 100 hours. Fast octadecane degraders, which had high cell hydrophobicity, were slightly inhibited by high levels of rhamnolipid, while for slow octadecane degraders with low cell hydrophobicity, rhamnolipid stimulated degradation. In Zhang and Miller (1995) it was reported that dirhmnolipids could enhance hydrocarbon dispersion and biodegradation as well as monorhamnolipids but at much lower concentrations (0.01 mM). Zhang and Miller speculated that in situ stimulation of dirhamnolipid concentrations sufficient to foster bioremediation might be feasible. Herman et al. (1997a) studied the effect of rhamnolipid on cell aggregation. One *Pseudomonas* strain studied formed extensive multicell aggregates in the presence of rhamnolipid at concentrations below the CMC. This aggregation greatly decreased the ability of the bacteria to degrade hexadecane in a sand packed column system. The other *Pseudomonas* strain did not aggregate, and biosurfactant enhanced in situ mineralization of hexadecane. In another study of the effects of attached bacteria and biosurfactants, Herman et al. (1997b) concluded that, in sand columns, bacteria biomass production had no effect on hexadecane immobilization. Herman et al. observed that a 0.1 mM rhamnolipid concentration had a minor effect on hexadecane mobilitization but that a 1.0 mM concentration significantly enhance mineralization. They suggested that the application of a biosurfactant following a period of biodegradation could be a useful remediation strategy. Herman et al. (1997b) noted in sand column studies that sand heavily contaminated with hexadecane (19.0% of pore volume) biodegradation with a specific *Pseudomonas aeruginosa* strain could remove 50% of the hexadecane by biodegradation alone, but that elution with 1 mM rhamnolipid could increase total mineralization to 65%.

Bai et al. (1997a) observing the influence of a rhamnolipid on the transport of bacteria though a sandy soil, found that the biosurfactant significantly increased the transport of *Pseudomonas aeruginosa* cells by reducing the irreversible adsorption of cells to soil particles. Bai et al. (1997b) compared at equal concentrations of 500mgl-1 a monorhamnolipid, Tween 80, and sodium dodecyl sulfate (SDS) for the removal of hexadecane from sand columns. Rhamnolipid removed 20% of the hexadecane compared to 6% for the Tween 80 and 0% for the SDS.

Zhang et al. (1997) compared the abilities of a monorhamnolipid and a dirhamnolipid to enhance the biodegradation of phenanthrene in an aqueous system. Monorhamnolipid was more effective for solubilization while the dirhamnolipid micelles were more bioavailable, so for both surfactants the rate of phenanthrene degradation was about the same.

An issue of concern with the use of rhamnolipid biosurfactants is that they are effective at complexation of heavy metals, which might be useful in soil flushing (Miller 1995b). This complexation could cause metals leaching during soil composting. This ability of rhamnolipid to release soil bound cadmium was confirmed by Torrens et al. (1998). Torrens et al. noted in their study that while rhamnolipid led to cadmium release from most soils, in high clay soil rhamnolipid cause soil dispersion and column plugging.

It needs to be noted that almost all surfactant studies are based on simplified laboratory systems, and that field investigative results are lacking. There appear to be no reports on the use of surfactants in soil composting or even in landfarming. Considering the great complexity of interaction among surfactants and petroleum hydrocarbons, soils, and soil microorganisms, it might take many years to determine if surfactant use will be an efficient and cost effective means of improving bioremediation efficiencies. Additionally, the complexities of petroleum hydrocarbon composition, soil type, microbial population structure, and other environmental factors might require the demonstration of surfactant enhancement of degradation on a case by case basis.

5. Methods development including tests for biodegradability, and analytical and instrumental investigative techniques

A number of tools and methods which may have application in the bioremediation of petroleum contaminated soils have been described in recent publications.

For the delineation of petroleum contaminated soils with near real time mapping, Kram et al. (1996), Kram et al. (1997), Kram (1998) and Kram and Lory (1998) have described the use of the US Navy Site Characterization and Analysis Cone Penetrometer System (SCAPS). The SCAPS system is based on the use of a small diameter cone penetrometer fitted with transducers for measuring point penetration resistance and a sapphire window connected to a pulsed nitrogen 337-nm laser and detector system at the surface. The SCAPS probe can derive soil classification information based on penetration resistance and can detect petroleum, oil and lubricants down to 100 ppm or lower via the laser detection system. A physical limitation of the SCAPS system is that it consists of a 20 ton vehicle of 10 metres length.

Some new tools have been described that could be of use for evaluation and monitoring of soil composting systems. VanderGheynst et al. (1997a) recounted a new type of in situ oxygen sensor that could be used in soil composting environments. Oppenheimer et al. (1997) described the use of a gas pycnometer for the determination of air filled porosity in unsaturated organic matrices such as compost. Determination of air filled porosity is a useful process control parameter because of the role of air filled porosity in oxygen and heat transfer.

VanderGheynst et al. (1997b) reported on the demonstration of a large (770 litre) pilot scale reactor used to mimic the central core of an aerated static bed or in-vessel composting system. Not clear in this work was why the pilot reactor was ventilated at various fixed rates, instead of time variable ventilation linked to microbial heat production as is used in most large scale high rate composting systems. VanderGheynst et al. (1997c) also developed a mathematical composting model and used the pilot system described in VanderGheynst et al. (1997b) to validate the mathematical model. Such models have predictive application to soil composting systems.

Reemtsma and Mehrtens (1997) described the determination of PAH leaching from contaminated soil using a column test with on-line solid phase extraction. A short cartridge of Supelclean LC-18 solid phase material was used for the on-line solid phase extraction of eluted PAHs. This method was described as good for the determination of leachable PAHs in soils, including recovery of PAHs at low (microgram) concentrations.

Aelion et al. (1997) used radiocarbon (14°C) measurement as a method of directly quantifying the evidence of in situ aerobic petroleum hydrocarbon biodegradation in soils and groundwater. This method would not work in soil composting because amendment with fresh organic matter would add significant 14C. Selifonov et al. (1998) have reported on the use of 13°C nuclear magnetic resonance spectroscopy with [1-13°C]Acenaphthene to study the bacterial degradation of acenaphthene in creosote/ PAH mixtures. As a point of interest, this techniques may also have potential for studying the role of humification in bound residue formation (Dec and Bollag 1997). It has been used successfully to study glucose incorporation into organic matter (Baldock et al 1990).This powerful method permits the direct identification of intermediate metabolites of acenaphthene degradation. This method could be used to contribute to our understanding of biochemistry of PAH bioremediation. Mansuy et al. (1997) used gas chromatography/ isotope ratio mass spectrometry as a means of identifying the source of petroleum and petroleum products. This method is based on the observation that while the chemical composition of petroleum can change greatly in the environment due to evaporation, water-washing or biodegradation, the isotopic compositions of individual components do not change. Analysis of the isotope ratios of a series of components provides a useful `finger print' identifying the petroleum source.

Shen et al. (1998) were able to identify, from soil, hydrocarbon degrading bacteria using a reverse sample genome probing (RSGP), and identification by 16S rDNA sequencing. The RSGP format permitted the rapid tracking of the abundance of multiple microbial genomes. This technique could be useful in tracking hydrocarbon degrading populations in soil composting investigations. Thorn et al. (1996) presented a method of isolating saprophytic basidiomycete fungi from soil, with the intent that this method could be useful for investigations of biodiversity and the isolation of organisms useful in bioremediation.

Methods for the testing of biodegradability of petroleum hydrocarbons have been recently considered. Pagga (1997) has reviewed the standard conditions for aerobic biodegradation tests, and considered testing in regard to the various ISO and OECD protocols used within the European Community. Pagga includes in his review an overview of standardized methods for determining biodegradability in water and compost. Tabak et al. (1997) described the development of a protocol for the determination of the kinetics of bioavailability and biodegradation for the purpose of establishing a framework for the enhancement of in situ soil remediation. Tabak et al. described an extensive mathematical model with validation derived from three types of model bioreactors. Battersby and Morgan (1997) demonstrated that the European test method, Biodegradability of Two-stroke Cycle Outboard Engine Oils in Water: Approved Test Method CEC L-33-A-93, could be used to predict the potential biodegradation of mineral oil based lubricants in soil. Comparing the results from the CEC test and a standard biodegradation test in soil microcosms gave highly similar results, indicating that the CEC method might be of value as a simple, quick and inexpensive test.

6. Studies of the degradation of specific compounds by identified cultures

Degradation of PAHs and other petroleum hydrocarbon fractions is carried out by a diverse number of microorganism with various degradative abilities. Understanding more about the role of specific populations can lead to insights as to how to environmentally select for appropriate degrader populations during soil remediation.

Studies of fungal degradation

Fungal degradation of PAHs especially have been of interest because of the enzymatic abilities of a number fungi to attack and degrade PAHs with higher numbers of aromatic rings. Sack et al. (1997a) compared the abilities of a range of wood decay fungi to degrade phenanthrene and pyrene. In liquid culture only *Trametes versicolor* and *Kuehneromyces mutabilis* were able to mineralize some fraction of phenathrene and pyrene. Yet in straw cultures (a very similar environment to composting) *T. versicolor* and *K. mutabilis* were able to mineralize more phenanthrene and pyrene. *Flammulina velutipes, Laetiporus sulphureus* and *Agrocybe aegerita*, which could not mineralize phenanthrene or pyrene at all in liquid culture, were able to mineralize significant amounts in straw culture. Sack et al. proposed that adsorption of PAHs onto straw might make the PAHs more bioavailable, and that growth on straw might induce the production of appropriate enzymes. Sack et al. (1997b) also investigated the ability of an *Aspergillus niger* strain, isolated from a petroleum contaminated soil, to degrade phenanthrene and pyrene. They found that *A. niger* produced two novel metabolites, 1-methoxyphenanthrene and 1-methoxpyrene, indicating yet another degradative pathway.

McFarland and Qui (1995) investigated the removal of benzo[a]pyrene in soil composting systems amended with the white rot fungus *Phanerochaete chrysosporium*. They found that compost amendment significantly enhanced benzo[a]pyrene removal, and the addition of *P. chrysosporium* did not further enhance removal.

Bezalel et al. (1996a,b) observed the mineralization of a number of PAHs by the white rot fungus *Pleurotus ostreatus*. Of note, there was no correlation between the activities of the lignolytic enzymes of *P. ostreatus* and the degradation of the PAHs, suggesting that membrane bound and intracellular enzymes were actually responsible for PAH initial degradation. Barclay et al. (1997), working with the white rot fungus *Phanerochaete chrysosporium* found that the mineralization of phenanthrene and benzo[a]pyrene by this fungus did not correlate with the production of lignin peroxidase activity, and that other intracellular enzymes appeared to be involved. Barclay et al. noted that the most common fate of phenanthrene was sorption and uptake by the fungus and subsequent transformation in to fungal biomass.

Bogan and Lamar (1996) looked at the PAH degrading abilities of *Phanerochaete laevis*, and found it was able to extensively degrade a number of common PAHs via an extra cellular manganese peroxidase mechanism. The intermediate metabolites produced by *P. laevis* were different from those observed produced by *P. chrysosporium*, indicating that generalities about degradative pathways for PAHs by *Phanerochaete* can not be made at the genus level. Hofrichter et al. (1998) considered the extracellular manganese peroxidase (MnP) system from the white rot fungus *Nematoloma frowardii*, and found that the cell free MnP system was able to partly mineralize to CO_2 a broad range of aromatic and aliphatic compounds. Hofrichter et al. concluded that this enzymatic combustion could represent an important pathway in the decomposition of various xenobiotics.

Ruttimann-Johnson and Lamar (1996) investigated the outcome of the effect of extracellular lignin degrading enzymes from P. chrysosporium on a phenolic compound, and observed the formation of high molecular weight polymers. Incorporation into humic organic matter might be an important removal mechanism in soil composting methods, related to the significant fungal activity, which occurs in these systems. May et al. (1997) used *Phanerochaete chrysosp*orium in a fungal reactor system to treat a soil highly contaminated with PAHs. Over 60 days, they observed that while only 2.5% of the total PAH concentration was mineralized, over 45% of the PAHs became polymerized within the soil via a quinone pathway. The potential for white rot fungi to polymerize PAHs in soil is significant in that these polymers have a much reduced toxicity and bioavailability compared to the original PAHs.

Studies of bacterial degradation

Whyte et al. (1997) isolated two psychrotrophic *Pseudomona*s sp. from petroleum contaminated Arctic soils, and found that both these strains possessed both the alk pathway for degrading alkanes and the nah pathway for PAH degradation. These catabolic pathways are found on different plasmids, but these strains contained both plasmids. These strains could readily metabolize various petroleum fractions at 5°C in a soil environment. Casellas et al. (1997) investigated the metabolic pathways of fluorene degradation by an *Arthrobacter* sp. Casellas et al. were able to identify intermediates and show that the *Arthrobacter* strain could degrade fluorene by three different pathways, illustrating the biochemical diversity of this bacterium.

Sugiura et al. (1997) compared the biodegradation of four crude oil samples by an *Acinetobacter* sp. and by a petroleum degrading bacterial consortium called SM8. There was a significant difference in the biodegradability of the four different crudes. While the *Acinetobacter* could not degrade PAHs, the SM8 was able to degrade a wide variety of PAHs with good efficiency. Rabus and Widdel (1996) demonstrated that four different denitrifying bacteria could utilize alkylbenzenes from petroleum as sole carbon sources under anoxic conditions. Kropp et al. (1997) investigated the biotransformations of the sulfur containing dimethldibenzothiophenes, which tend to be one of the more degradation resistant classes of common petroleum compounds. While a *Pseudomonas* sp. was rather limited in ability to degrade different dimethyldibenzothiophenes, two of the mixed culture populations were able to degrade to some extent all of the dimethyldibenzothiophenes.

7. Microbial Ecology of Petroleum Bioremediation

Komukia-Nakamura et al. (1996) examined the biodegradation of crude oil by a constructed consortia of four bacteria. *Rhodococcus* sp. and *Sphingomonas* sp. would not grow as part of the consortia. *Acinetobacter* sp. grew and degraded alkanes and other hydrocarbons, and produced metabolites that fostered the growth of Pseudomonas putida, which then degraded various aromatic compounds. Bouchez et al. (1995) tried various combinations of six bacterial strains able to utilize at least one PAH out of a group of six commonly found PAHs. Inhibitions and synergism's were observed among strains. Naphthalene was found to be toxic to all strains not isolated on this compound, so that the inclusion of a naphthalene degrading bacteria was required by non-naphthalene degrading bacteria in consortia.

Steward et al. (1997) assayed the distribution of oil tolerant and oil degrading bacteria around the site of a natural crude oil seepage. They found that there was a good deal of community diversity even over a small area even with the essentially the same environment, and that representative sampling of petroleum degrading bacteria in the natural environment needs to take this into account. Kireeva and Vodopyanov (1996a) looked at the declines in various microbial population and soil enzyme activities associated with petroleum contamination of soil. While dose dependency limited the use of the model to predicting the affect of petroleum pollution on populations, changes in populations observed during the first year of pollution could be used to determine the initial dose of oil pollution.

Margesin and Schinner (1997d) considered the influence of temperature on diesel oil degradation in mineral medium and in soil. While the psychrotrophic yeast *Yarrowia lipolytica* degraded diesel well between 10-20 °C in liquid culture, the yeast required a higher temperature and degraded diesel much more poorly in soil. Indigenous soil microorganism degraded diesel at all tested temperatures and over time displaced the inoculated yeast. In further work Margesin and Schinner (1997b) looked at stimulating bioremediation of diesel oil in a cool alpine soil. While fertilization at a C/N/P ratio of 100:10:2 greatly enhanced degradation, inoculation with a psychrotrophic diesel degrader with or without fertilization had no effect.

Coates et al. (1997) reported that under strict anaerobic conditions in bay sediments, many PAHs derived from petroleum hydrocarbons could be mineralized under sulfate reducing conditions. This is of note in that gypsum is commonly added to soils to improve aggregate structure, and that similar sulfate reducing conditions could occur within gypsum amended soil composting piles. That the same sulfur reactions can occur in marine and composting ecosystems has been demonstrated (Miller 1993b). Weiner and Lovely (1998) also looked at petroleum contaminated sediments where benzene persisted, and surmised that the lack of degradation was caused by a failure of the appropriate sulfate reducers to colonize the sediments. In a three years study of crude oil contaminated soils, with or without fertilizer addition, Duncan et al. (1997) did find up to a 1000 fold increase in sulfate reducing bacteria in contaminated soils as compared to controls. Duncan et al. also found that oil contamination significantly increased the abundance of fungi especially in upper soil layers.

The fate and activity of microorganisms introduced into soil has been recently reviewed by vanVeen et al. (1997). Generally, inoculations into soil fail, and the reason `is the general obstinacy of the soil ecosystem, which normally acts as a buffer against incoming microorganisms.' Only in the case of strong ecological selective pressures in favor of the inoculant do inoculants occasionally become successfully introduced. These observations fully apply to soil bioremediation situations. Kastner et al. (1998) inoculated soil containing PAHs with PAH degrading bacteria. They found that the carrier in which the bacteria were applied, such as a mineral salts medium, had a great effect on both the survival of the inoculant and on the activity of indigenous PAH degraders. In earlier work Kastner and Mahro (1996) had amended this same soil with compost and found that compost addition significantly improved PAH degradation.

Zogg et al. (1997) investigated the compositional and functional shifts in microbial communities caused by soil warming. Their results indicate not only does the rate of decomposition increase with moderate increases in temperature, but increasing temperature causes population shifts leading to differential access to substrate pools. As soil temperature increased from 5°C to 25°C, total active microbial biomass decreased by half, while carbon turnover increased 3 times.

II B. Properties of Composted Materials and End Use

8. Environmental fate of petroleum products and metabolic intermediates

One of the most active groups investigating the fate of PAHs in soils has been that of Steinhart and colleagues at the Universitat Hamburg. Langbehn and Steinhart (1994) analyzed the organic acids and ketones formed during the biodegradation of diesel and lubricating oils in soil and were able to quantify dozens of acids and ketones. Langbehn and Steinhart (1995) continued this work and identified a quite extensive list of intermediate organic compound believed to be products of the biodegradation of diesel and lubricating oils. They noted that while in the laboratory cyclic carboxylic acids were broken down in weeks, these same acids tended to persist for years in actual contaminated soils. Even aromatic and branched aliphatics decomposed in laboratory studies whereas minimal breakdown occurred in actual soils. Wischmann et al. (1997) developed an extraction procedure that permits analysis of PAHs and metabolites from soil/ compost samples. In an eleven week degradation study of soil mixed with compost on a 9:1 ratio, phenanthrene was highly degraded, 33% of the fluorene was degraded, and fluoranthene and benz[a]anthracene decreased insignificantly. Intermediate metabolite products, 1-hydroxy-2-naphthoic acid for phenanthrene, and 9-fluorenol and 9-fluorenone for fluorene, were degraded and did not accumulate in the biotic soil/ compost mixture, but persisted in a sterile soil/ compost mixture.

Wischmann and Steinhart (1997) investigated the degradation of 18 PAHs (some containing nitrogen) in three soils and with and without compost amendment. In unamended soils only aromatics with up to three benzene rings were degraded, while in the compost amended soil all PAHs were either eliminated or substantially reduced within 15 weeks. In the amended soil the only remaining residues were 11%, 19% and 54% of the benz[a]anthracene, chrysene and benzo[a]pyrene, respectively. Further, in the amended soil, intermediate products of PAH decomposition did not accumulate, although it could not be determined if this reflected further degradation or fixing within the soil organic matrix. Of note, in sterile soil systems, abiotic oxidation of PAHs occurred to an appreciable extent, leading to the accumulation of various oxidized products. Considering the role of organic matter in bioremediation, Vecchioli et al. (1997) found in studies of field application of petroleum hydrocarbons that soil organic matter adsorption capacity was the factor in determining hydrocarbon elimination limits in soil bioremediation.

Kanaly et al. (1997) used radiolabelled benzo[a]pyrene added to soil along with crude oil to study the fate of this PAH. After 150 days only 5% of the benzo[a]pyrene remained, with 50% label in the CO2 evolved, 20% in solvent extractable metabolites, and 25% incorporated, in a mostly unextractable form, into the soil organic matter. They concluded that while benzo[a]pyrene persists in the environment because it cannot support microbial growth, in the presence of suitable cosubstrates it can be completely degraded. Anderson and Henrysson (1996) compared the abilities of the white rot fungi *Trametes versicolor, Pleurotus ostreatus, P. sajor-caju*, and *Phanerochaete chrysosporium* to degrade the PAHs anthracene, benz[a]anthracene and dibenz[a,h]anthracene in soil. While Trametes degraded the PAHs with little or no accumulation of dead-end metabolites, Phanerochaete and Pleurotus almost completely converted anthracene to non-degradable 9,10-anthracenedione. Comparisons between soil and liquid culture data gave quite different results.

Saeed et al. (1998) reported the results of a long term study on the change in composition with time of various petroleum's in Kuwaiti oil lakes resulting from the Gulf War of 1991. During weathering, the fraction of asphaltenes increased considerably, saturates decreased considerably for 21 months and then increased, aromatics decreased except for the higher weight PAHs, and resins increased.

Rhykerd et al. (1998) investigated the loss of low molecular weight volatiles from freshly spill crude oil on to soil, and found that organic amendments such as sawdust, hay and peat could significantly reduce volatile losses. Unfortunately, volatile losses tend to occur so quickly that it might not be practical to add adsorbent agents quickly enough.

9. End usage including issues of risk and effects on soil and plant growth

Weisman (1998) has described the efforts of the `Total Petroleum Hydrocarbon Criteria Working Group' in developing a risk-based approach for the management of total petroleum hydrocarbons in soils. The Working Group has more than 400 participants from industry, government and academia. The goal of the Working Group is develop scientifically sound human risk based standards for site remediation. Five volumes are either in recent publication, in press, or in preparation from the Working Group, and other information will soon be available at the web site (http://voyager.woafb.af.mil).

Li et al. (1997) studied the growth of barley in crude oil contaminated soil and in the same soil after bioremediation. They found that bioremediation did not improve barley growth, because bioremediation did not address the problem of poor soil water sorption caused by the hydrocarbon residuals. Conclusions were that soil hydrophobicity caused by petroleum contamination was a major problem and that the soil/ water relationship should be one of the most important factors in assessing the endpoint for bioremediation. Sawatsky and Li (1997) compared a petroleum contaminated soil, the same soil after three years of bioremediation and a control. Above a critical water content of 18% for the bioremediated soil, and 20% for the contaminated soil, sorptivity was near to the control soil. Below these critical values there was a strong divergence in sorptivity. After air drying, the contaminated and bioremediated soil exhibited a water infiltration rate 3 orders of magnitude slower than the control soil.

While Li et al. (1997) and Sawatsky and Li (1997) found that petroleum had an adverse affect on soil properties, the application of biodegradable vegetable oils does not. Plante and Voroney (1998) investigated the affect of land application of oily food waste and canola oil on soil properties. In contrast to petroleum, oily food waste and canola oil degrade rapidly and actually improve soil structure by improving soil aggregate stability.

A problem associated with crude oil contamination of soils is that such contamination can often be accompanied by contamination with brines. In high clay soils this is a special problem, in that sodium can disperse clay colloids and leave soil lacking in structure and pore space. Barzegar et al. (1997) investigated the issues of organic matter content, sodicity, and clay type on soil aggregation. They demonstrated that added soil organic matter can greatly decrease clay dispersion and improve soil aggregation, and that organic matter can have this effect as well in sodic soils as non sodic soils. These results would indicate that the organic matter added to soils during soil composting should significantly improve the structure of high clay soils even in the presence of higher salt levels.

Chaineau et al. (1997) grew lettuce, clover, barley, bean, sunflower, and maize on fuel oil contaminated soils. Wheat and bean were greatly stunted by fuel oil, while corn was relatively much less affected. Even with high dose of fuel hydrocarbons, however, fuel hydrocarbons were not detected within aerial plant biomass. Leyval and Binet (1998) treated soil with anthracene to evaluate the effect of PAHs on arbuscular mycorrhizal fungi. Effects of PAHs on plant survival and growth depended on species, in that for some plants mycorrhizal colonization decreased, while for maize and ryegrass colonization was not affected. At 5 g of PAH kg-1 soil, only mycorrhizal plants survived.

Plant growth on bioremediated soils is important for many reasons, including that plant growth itself can further remediate petroleum contamination. Wiltse et al. (1998) found alfalfa can be used to phytoremediate crude oil contaminated soils, but that different genotypes lead to significantly different results in the extent of oil degradation.

III. POSITION PAPER

General considerations

Remediation of petroleum contaminated soils is obviously a topic of significant academic interest. Soil composting methods are of significant private and public sector interest with a good deal of field work being undertaken, based on conversations with individuals in industry, government, and trade organizations such as the Canadian Association of Petroleum Producers and the American Petroleum Institute. Yet, there is an apparently large gap between academic research and actual field remediation work, evidenced by the real lack in the scientific literature of actual field scale soil composting remediation studies. In the scientific literature there is little research that can be directly applied to field soil composting application, while the few publications that deal directly with soil composting contain little scientific information on which to build a better understanding. To forward the technology of soil composting for the remediation of petroleum contamination, the gap between laboratory work and field application needs to be bridged.

While not documented, discussions with a number of scientists and industry people interested in the remediation of petroleum contaminated soils gives explanation of why there is a lack of published field research in soil composting, even while significant industrial money has been spent. Many of the larger companies in the oil industry with an interest in soil remediation perceive a proprietary interest in research areas of remediation. That is, to be able to successfully carry out bioremediation of soils while others can not, is viewed as offering proprietary advantage or, viewed as a leverage contribution opportunity. A number of studies have been carried out by university researchers for large oil companies with the provision that such research be kept confidential and never published (Richard Bartha, personal communication). This proprietary viewpoint does not advance the science and application of bioremediation, nor the public interest in environmental quality. It might also be argued that a proprietary viewpoint is not in the best interests of industry, as progress in soil composting methods over the past 20 years has been slow. Perhaps, it can best be advanced via industry, government, universities and college collaborative and jointly funded programs.

In considering how to advance the state of the art in soil composting, it is worthwhile to consider how progress has been made in waste composting. Progress in waste composting was hampered for many years by the false perception that composting was a natural process, and that therefore process management was of little concern. Millions of dollars of investor's and tax payer's money were spent on large facilities that did not work because of a lack of process control. In the later 1970's and 1980's government funded basic and applied research was carried out that permitted the composting ecosystem to be understood in reference to waste treatment goals, allowing for the development of rational process control. A fundamental ecological understanding of process control has lead to the current success of waste composting. While the private sector has contributed to improvements in materials handling (Haug 1993), the advancements in composting process management have come from academic research (Miller 1993a). While much knowledge of waste composting can be applied to soil composting, soil composting requires its own strategies for process control to address the goal of remediating compounds that can be recalcitrant and potentially toxic.

As was brought forth in the Summary of Introduction of Composting as a Remediation Process (provided in the original proposal of October 1997 to AEP), bioremediation is a process approachable through ecosystem management. Many aspects of the composting ecosystem can be managed, including texture and porosity, nutrient content, moisture, oxygen concentration, temperature, pH, redox potential, and other parameters. It has been amply demonstrated that management of various ecological factors can tremendously affect both the course and final outcome of composting (Miller 1993). Yet, as the current literature demonstrates, in field application soil composting is not only not managed as an ecological process, but field data even describing general environmental conditions within soil composting materials are lacking. The paper of Al-Daher et al. (1998) and the US EPA report (US EPA 1998) are typical examples of the `black box' approach to soil composting, in which no management of processing conditions, or even a description of processing conditions, is attempted. While from an ecological perspective a good deal is known about composting from field and laboratory studies, and a fair amount of ecological information on soil remediation can be derived from laboratory studies, to date there is still a great lack of information about the ecology of field soil composting.

Technical issues brought forth in the literature review

A major technical issue is that to date there is a great lack of ecological data from either soil composting remediations that worked or from ones that did not work. We know a fair amount of ecological information about specific petroleum degraders or degrading consortia, but this understanding can not be applied to the field because there is no field data to interpret. Field data is sparse for the effects of temperature, matric and osmotic water potential, oxygen, redox potential, pH and a host of other factors. About the only topic for which there is a fair amount of data is for C:N or C:N:P:K ratios.

Clean up levels are an ongoing technical and regulatory issue. Sequestration of petroleum fractions into soil mineral, and especially organic matter, can greatly decrease mobility and bioavailability. As was expressed by Alexander (1995) and reiterated by a number of authors since, what is the environmental risk of a PAH or other hydrocarbon that has little or no bioavailability? This is not to discount the important observations by Li et al. (1997) and Sawatsky and Li (1997) that petroleum contamination of soils can have a persist adverse effect on soil/water relations. The soil matrix provides the habitat for organisms that carry out ecological functions such as carbon and nitrogen fixation, nutrient cycling, and decomposition process. The soil habitat must provide “living” space as well as air, water, and nutrient. The effects of hydrocarbons on the soil ecosystem are twofold: 1. Hydrocarbons may be toxic to the organisms inhabiting or otherwise contacting soil; and 2. Hydrocarbons may impart undesirable physical characteristics that impair the soil’s ability to serve as habitat for these organisms. Remediation objectives must protect the soil’s ability to fulfill its role within the ecosystem. In order to do so, they must protect against both toxic effects (bioavailability plays a role here) and impairment of physical characteristics. Clear in the literature is that there is a difference between what can be extracted from the soil by harsh chemical extractions, and what is bioavailable.

Current literature confirms what has generally been believed by microbial ecologists for some time regarding inoculants. Sites that have been contaminated for some time tend to have diverse and well developed populations of petroleum degraders (Steward et al. 1997), but amendment with adapted seed cultures is useful when situations have prevented the development of degrader populations (Guthrie and Pfaender (1998). While a great deal of work has been carried out with various strains of bacteria and fungi known to be petroleum hydrocarbon degraders, there is still a lack of any evidence that use of special degrader inoculants would be of any value in soil remediation.

Soil organic matter, and even more recently introduced organic matter, seems to have an active role in the fate of petroleum hydrocarbons in soil. Organic matter not only can bind hydrocarbons and decrease bioavailability, but apparently soil organic matter along with soil minerals can catalyze abiotic oxidation's and other reactions of petroleum compounds. As soil composting methods usually introduce organic matter, the function of introduced organic matter needs to be better understood.

Suggestions for further research

To realize the potential for soil composting as a valuable method of remediating petroleum contamination of soils, further research is warranted. Following are ideas for further research derived from the literature review.

1. Based on the literature, there are only somewhat vague ideas as to optimal ranges of environmental parameters to be achieved during soil composting. While the optimal temperature range for waste composting is 55-60°C, the optimal range for petroleum remediation is lower. Research of Hogan (1994) indicates that for PAH degradation there are actually rather different optimal temperatures for the degradation of specific PAH's. This raises the question as to whether there is no single optimal temperature, but rather different temperatures at different stages of processing which might optimize activity. Along these lines, determination of optimal oxygen concentration, pH, redox potential, and matric and osmotic potential also require further inquiry. Populations responsible for petroleum biodegradation will only carry out their role when environmental conditions are made selective for their growth and desired activity. While a great deal of investigation into the biodegradation of petroleum and PAH's has been carried out with pure cultures, in actual application soil composting will occur through consortia of many bacteria and fungi. More work, like that of Sugiura et al. (1997), needs to be carried out looking at the activity and means of environmental selection for consortia of organisms that can successfully remediate petroleum.

2. A wide range of environmental conditions within soil composting windrows or piles have yet to be described. It is impossible to improve a process in a situation where very little is understood about processing. What is known about bioremediation of petroleum, and composting in general, can not be used to interpret or improve a process when there is a great lack of information on process ecology. Detailed modeling of composting processes should consider soils, bulking agents, weather, and site conditions as these parameters very significantly in field cases.

3. Surfactants have been of substantial research interest of late, because surfactants have been demonstrated to improve the bioavailability of PAH's and other petroleum fractions in laboratory studies. Whether or not surfactant addition could be of benefit in field application has not been addressed. Surfactants studies to-date have produced ambiguous results at best. (Nicholson personal communication). Evaluation of the most promising surfactants and their ecological effects in field application would be useful.

4. Production of surfactants by bacteria in the presence of PAH's has been recently demonstrated by Deziel et al. (1996). Methods exist to investigate surfactant production in soil and composting matrices. It would be worthwhile to investigate the extent to which biosurfactants are being produced during soil composting, because this could significantly affect the extent of remediation. Ecological investigation into in-situ biosurfactant production could also be useful, in that environmental management might be able to be used to select for conditions that will promote biosurfactant production and therefore enhance bioavailability. Understanding the processing conditions, which enhance biosurfactant production, could lead to improved models of process management.

5. Soil organic matter appears to have an active role in the degradation of petroleum hydrocarbons (Kastner and Mahro 1996, Sack et al. 1997a, Wischmann and Steinhart 1997), and also a role in sequestration. A question yet to be addressed in soil composting is how much benefit is derived from composting activity induced by the addition of organic matter versus the effect of organic matter addition alone. Could amending petroleum contaminated soil with finished compost work as well as amendment with compostable organics? Amending soil with finished compost would likely be cheaper and simplify process management.

6. As pointed out by Li et al. (1997) and Sawatsky and Li (1997), a difficulty in the remediation of petroleum contaminated soil is that of residual hydrophobicity and its adverse affect on soil/ water relations. Compost is well known for holding water and improving the ability of soils to wet, hold water, and provide water for plant growth. In bioremediated soils with residual hydrophobicity, could a compost amendment be used to overcome the hydrophobicity problem and permit the establishment of healthy plant growth, as well, will composting produce a soil that is free from hydrophobicity?

7. Plant growth in the soil can promote significant bioremediation of petroleum hydrocarbons (Wiltse et al. 1998), although the type of plant used is important because plants vary greatly in their response to petroleum contamination (Chaineau et al. 1997). Could a practical soil bioremediation program be based on an initial stage of soil composting, carried out sufficiently to support some plant growth, and followed by phytoremediation with an appropriate cover crop for further remediation?

8. Regulatory end points for remediation could likely be improved in reference to remediation costs while still protecting human health and the environment (Weisman 1998). Clean up based on chemical analysis standards that can not distinguish between normal soil aromatic compounds in humus and those derived from petroleum hydrocarbons place unwarranted restrictions on the very practices, such as composting and organic amendment, that would remediate the contamination.

Environmental risks from PAH's and other petroleum fractions that have been truly sequestered onto soil organic matter are greatly reduced (Alexander 1995), yet harsh chemical extraction procedures can still recover some fraction of sequestered compounds. Testing methods that reflect environmental availability might have value in determining clean up standards, however, more research is needed in the development of such methods. Endpoint determination remains a critical area for future research.

9. Addition of mineral nutrients has been known to be of value in the remediation of petroleum contamination in soils for quite a while. Yet, response to fertilizer amendment is not always as desired because determination of the optimal amount to use is soil dependent. Work of Walworth et al. (1997a,b) using soil water concentration as a basis for nutrient amendment seems quite logical. Further development of the ideas of Walworth et al. for routine field application might be worthwhile.

10. A strong response from industry and government has identified two research problems, which require further work:

The first is the development of testing methods that reflect bioavailability. The second is the development of testing methods that distinguish natural organic compounds from the contaminant. The latter question is especially relevant to compost where large quantities of naturally occurring organic materials (i.e. manure, straw, wood waste) are added during the remediation process. High concentrations of naturally derived organic matter interfere with some analytical methods for measuring hydrocarbons.

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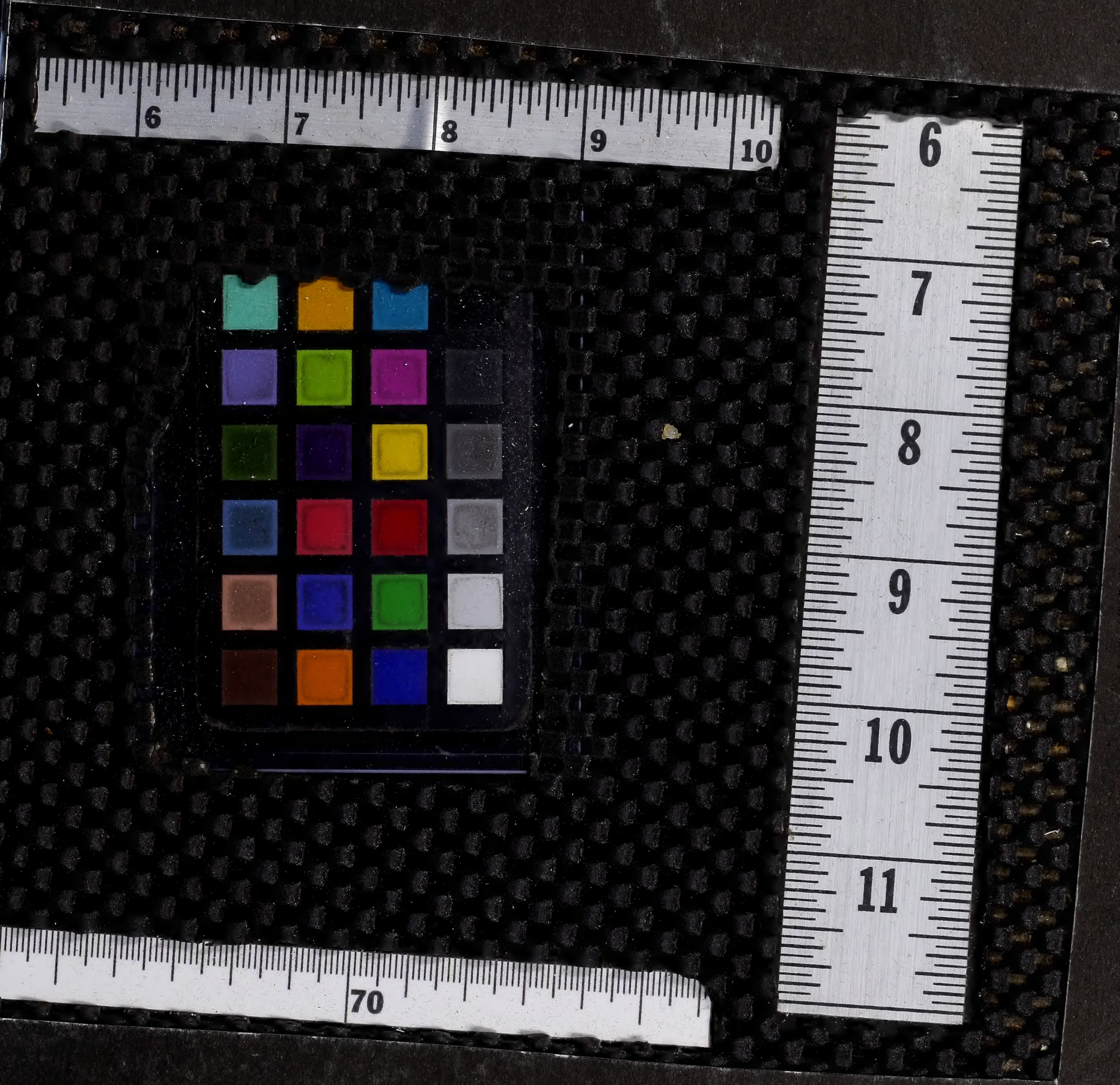
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